5. Agriculture

Agricultural activities contribute directly to emissions of greenhouse gases through a variety of processes. The Agriculture chapter includes the following sources: enteric fermentation in domestic livestock, livestock manure management, rice cultivation, agricultural soil activities, and agricultural residue burning (see Figure 5-1). Several other agricultural activities, such as irrigation and tillage practices, may also generate anthropogenic greenhouse gas emissions; however, the impacts of these practices are too uncertain to estimate emissions.[1] Agriculture-related land-use activities, such as conversion of grassland to cultivated land, are discussed in the Land-Use Change and Forestry chapter.

In 1997, agricultural activities were responsible for emissions of 131.4 MMTCE, or 7 percent of total U.S. greenhouse gas emissions. Methane (CH_4) and nitrous oxide (N_2O) were the primary greenhouse gases emitted by agricultural activities. Methane emissions from enteric fermentation and manure management represent about 19 and 9 percent of total CH_4 emissions from anthropogenic activities, respectively. Of all domestic animal types, beef and dairy cattle were by far the largest emitters of methane. Rice cultivation and agricultural crop waste burning were minor sources of methane. Agricultural soil management activities such as fertilizer application and other cropping practices were the largest source of U.S. N_2O emissions, accounting for 68 percent. Manure management and agricultural residue burning were also smaller sources of N_2O emissions.

Figure 5-1

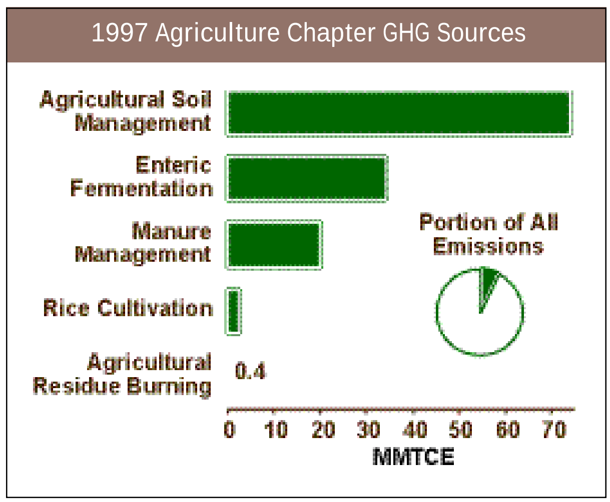

Table 5-1 and Table 5-2 present emission estimates for the Agriculture chapter. Between 1990 and 1997, CH_4 emissions from agricultural activities increased by 8 percent while N_2O emissions increased by 13 percent. In addition to CH_4 and N_2O, agricultural residue burning was also a minor source of the criteria pollutants carbon monoxide (CO) and nitrogen oxides (NO_x).

[1] Irrigation associated with rice cultivation is included in this inventory.

Table 5-1: Emissions from Agriculture (MMTCE)

Gas/Source	1990	1991	1992	1993	1994	1995	1996	1997
CH_4	**50.3**	**50.9**	**52.2**	**52.5**	**54.5**	**54.8**	**53.8**	**54.1**
Enteric Fermentation	32.7	32.8	33.2	33.6	34.5	34.9	34.5	34.1
Manure Management	14.9	15.4	16.0	16.1	16.7	16.9	16.6	17.0
Rice Cultivation	2.5	2.5	2.8	2.5	3.0	2.8	2.5	2.7
Agricultural Residue Burning	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2
N_2O	**68.1**	**69.1**	**70.9**	**69.9**	**76.4**	**73.2**	**75.1**	**77.2**
Manure Management	2.6	2.8	2.8	2.9	2.9	2.9	3.0	3.0
Agricultural Soil Management	65.3	66.2	68.0	67.0	73.4	70.2	72.0	74.1
Agricultural Residue Burning	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1
Total	**118.4**	**120.0**	**123.1**	**122.4**	**130.9**	**128.0**	**128.9**	**131.4**

Note: Totals may not sum due to independent rounding.

Table 5-2: Emissions from Agriculture (Tg)

Gas/Source	1990	1991	1992	1993	1994	1995	1996	1997
CH_4	**8.8**	**8.9**	**9.1**	**9.2**	**9.5**	**9.6**	**9.4**	**9.4**
Enteric Fermentation	5.7	5.7	5.8	5.9	6.0	6.1	6.0	6.0
Manure Management	2.6	2.7	2.8	2.8	2.9	3.0	2.9	3.0
Rice Cultivation	0.4	0.4	0.5	0.4	0.5	0.5	0.4	0.5
Agricultural Residue Burning	+	+	+	+	+	+	+	+
N_2O	**0.8**	**0.8**	**0.8**	**0.8**	**0.9**	**0.9**	**0.9**	**0.9**
Manure Management	+	+	+	+	+	+	+	+
Agricultural Soil Management	0.8	0.8	0.8	0.8	0.9	0.8	0.9	0.9
Agricultural Residue Burning	+	+	+	+	+	+	+	+

+ Does not exceed 0.05 Tg

Note: Totals may not sum due to independent rounding.

Enteric Fermentation

Methane (CH_4) is produced as part of the normal digestive processes in animals. During digestion, microbes resident in an animal's digestive system ferment food consumed by the animal. This microbial fermentation process, referred to as enteric fermentation, produces methane as a by-product, which can be exhaled, or eructated, by the animal. The amount of methane produced and excreted by an individual animal depends primarily upon the animal's digestive system, and the amount and type of feed it consumes.

Among domestic animal types, the ruminant animals (e.g., cattle, buffalo, sheep, goats, and camels) are the major emitters of methane because of their unique digestive system. Ruminants possess a rumen, or large "fore-stomach," in which microbial fermentation breaks down the feed they consume into soluble products that can be utilized by the animal. The microbial fermentation that occurs in the rumen enables ruminants to digest coarse plant material that non-ruminant animals cannot. Ruminant animals, consequently, have the highest methane emissions among all animal types.

Non-ruminant domestic animals (e.g., pigs, horses, mules, rabbits, and guinea pigs) also produce methane through enteric fermentation, although this microbial fermentation occurs in the large intestine. These non-ruminants have significantly lower methane emissions than ruminants because the capacity of the large intestine to produce methane is lower.

In addition to the type of digestive system, an animal's feed intake also affects methane excretion. In general, a higher feed intake leads to higher methane emissions. Feed intake is positively related to animal size, growth rate, and production (e.g., milk production, wool growth, pregnancy, or work). Therefore, feed intake varies among animal types as well as among different management practices for individual animal types.

Methane emissions estimates for livestock are shown in Table 5-3 and Table 5-4. Total livestock emis-

Table 5-3: CH_4 Emissions from Enteric Fermentation (MMTCE)

Animal Type	1990	1991	1992	1993	1994	1995	1996	1997
Dairy Cattle	**8.4**	**8.4**	**8.4**	**8.4**	**8.4**	**8.4**	**8.3**	**8.3**
Beef Cattle	**22.6**	**22.8**	**23.1**	**23.6**	**24.5**	**24.9**	**24.6**	**24.3**
Other	**1.6**	**1.7**	**1.7**	**1.6**	**1.6**	**1.6**	**1.6**	**1.6**
Sheep	0.5	0.5	0.5	0.5	0.4	0.4	0.4	0.3
Goats	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1
Horses	0.5	0.6	0.6	0.6	0.6	0.6	0.6	0.6
Hogs	0.5	0.5	0.5	0.5	0.5	0.5	0.5	0.5
Total	**32.7**	**32.8**	**33.2**	**33.6**	**34.5**	**34.9**	**34.5**	**34.1**

Note: Totals may not sum due to independent rounding.

Table 5-4: CH_4 Emissions from Enteric Fermentation (Tg)

Animal Type	1990	1991	1992	1993	1994	1995	1996	1997
Dairy Cattle	**1.5**	**1.5**	**1.5**	**1.5**	**1.5**	**1.5**	**1.5**	**1.5**
Beef Cattle	**4.0**	**4.0**	**4.0**	**4.1**	**4.3**	**4.3**	**4.3**	**4.2**
Other	**0.3**	**0.3**	**0.3**	**0.3**	**0.3**	**0.3**	**0.3**	**0.3**
Sheep	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1
Goats	+	+	+	+	+	+	+	+
Horses	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1
Hogs	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1
Total	**5.7**	**5.7**	**5.8**	**5.9**	**6.0**	**6.1**	**6.0**	**6.0**

+ Does not exceed 0.05 Tg
Note: Totals may not sum due to independent rounding.

sions in 1997 were 34.1 MMTCE (6.0 Tg). Emissions from dairy cattle remained relatively constant from 1990 to 1997 despite a steady increase in milk production. During this time, emissions per cow increased due to a rise in milk production per dairy cow (see Table 5-5); however, this trend was offset by a decline in the dairy cow population. Beef cattle emissions continued to decline, caused by the second consecutive year of declining cattle populations. Methane emissions from other animals have remained relatively constant.

Methodology

Livestock emission estimates fall into two categories: cattle and other domesticated animals. Cattle, due to their large population, large size, and particular digestive characteristics, account for the majority of methane emissions from livestock in the United States and are handled separately. Also, cattle production systems in the United States are well characterized in comparison with other livestock management systems. Overall, emissions estimates were derived using emission factors, which were multiplied by animal population data.

While the large diversity of animal management practices cannot be precisely characterized and evaluated, significant scientific literature exists that describes the quantity of methane produced by individual ruminant animals, particularly cattle. A detailed model that incorporates this information and other analyses of feeding practices and production characteristics was used to estimate emissions from cattle populations.

To derive emission factors for the various types of cattle found in the United States, a mechanistic model of rumen digestion and animal production was applied to data on thirty-two different diets and nine different cattle types (Baldwin et al. 1987a and b).[2] The cattle types were defined to represent the different sizes, ages, feeding systems, and management systems that are typically found in the United States. Representative diets were

[2] The basic model of Baldwin et al. (1987a and b) was revised somewhat to allow for evaluations of a greater range of animal types and diets. See EPA (1993).

defined for each category of animal, reflecting the feeds and forages consumed by cattle type and region. Using this model, emission factors were derived for each combination of animal type and representative diet. Based upon the level of use of each diet in the five regions, average regional emission factors for each of the nine cattle types were derived.[3] These emission factors were then multiplied by the applicable animal populations from each region.

For dairy cows and beef cows and replacements, emission estimates were developed using regional emission factors. Dairy cow emission factors were modified to reflect changing (primarily increasing) milk production per cow over time in each region. All other emission factors were held constant over time. Emissions from other cattle types were estimated using national average emission factors.

Emissions estimates for other animal types were based upon average emission factors representative of entire populations of each animal type. Methane emissions from these animals accounted for a minor portion of total methane emissions from livestock in the United States. Also, the variability in emission factors for each of these other animal types (e.g., variability by age, production system, and feeding practice within each animal type) is smaller than for cattle.

See Annex G for more detailed information on the methodology and data used to calculate methane emissions from enteric fermentation.

Data Sources

The emission estimates for all domestic livestock were determined using a mechanistic model of rumen digestion and emission factors developed in EPA (1993). For dairy cows and beef cows and replacements, regional emission factors were used from EPA (1993). Emissions from other cattle types were estimated using national average emission factors from EPA (1993). Methane emissions from sheep, goats, pigs, and horses were estimated by using emission factors utilized in Crutzen et al. (1986) and annual population data from USDA statistical reports. These emission factors are representative of typical animal sizes, feed intakes, and feed characteristics in developed countries. The methodology employed in EPA (1993) is the same as those recommended in IPCC (1997). All livestock population data were taken from USDA statistical reports. See the following section on manure management for a complete listing of reports cited. Table 5-5 below provides a summary of cattle population and milk production data.

Table 5-5: Cow Populations (thousands) and Milk Production (million kilograms)

Year	Dairy Cow Population	Beef Cow Population	Milk Production
1990	10,007	32,677	67,006
1991	9,883	32,960	66,995
1992	9,714	33,453	68,441
1993	9,679	34,132	68,304
1994	9,514	35,325	69,702
1995	9,494	35,628	70,500
1996	9,409	35,414	69,976
1997	9,304	34,486	71,035

Uncertainty

The diets analyzed using the rumen digestion model include broad representations of the types of feed consumed within each region. Therefore, the full diversity of feeding strategies employed in the United States is not represented and the emission factors used may be biased. The rumen digestion model, however, has been validated by experimental data. Animal population and production statistics, particularly for beef cows and other grazing cattle, are also uncertain. Overall, the uncertainty in the emission estimate is estimated to be roughly “20 percent (EPA 1993).

Manure Management

The management of livestock manure produces methane (CH_4) and nitrous oxide (N_2O) emissions. Methane is produced by the anaerobic decomposition of manure. Nitrous oxide is produced as part of the agricultural nitrogen cycle through the denitrification of the organic nitrogen in livestock manure and urine.

[3] Feed intake of bulls does not vary significantly by region, so only a national emission factor was derived for this cattle type.

When livestock and poultry manure is stored or treated in systems that promote anaerobic conditions (e.g., as a liquid in lagoons, ponds, tanks, or pits), the decomposition of materials in manure tends to produce methane. When manure is handled as a solid (e.g., in stacks or pits) or deposited on pastures and range lands, it tends to decompose aerobically and produce little or no methane. Air temperature and moisture also affect the amount of methane produced because they influence the growth of the bacteria responsible for methane formation. Methane production generally increases with rising temperature and residency time. Also, for non-liquid based manure systems, moist conditions (which are a function of rainfall and humidity) favor methane production. Although the majority of manure is handled as a solid, producing little methane, the general trend in manure management, particularly for dairy and swine producers, is one of increasing usage of liquid systems.

The composition of the manure also affects the amount of methane produced. Manure composition depends upon the diet of the animals. The greater the energy content and digestibility of the feed, the greater the potential for methane emissions. For example, feedlot cattle fed a high energy grain diet generate manure with a high methane-producing capacity. Range cattle feeding on a low energy diet of forage material produce manure with only half the methane-producing capacity of feedlot cattle manure.

The amount of N_2O produced can also vary depending on the manure and urine composition, the type of bacteria involved in the process, and the amount of oxygen and liquid in the manure system. Nitrous oxide emissions result from livestock manure and urine that is managed using liquid and slurry systems, as well as manure and urine that is collected and stored. Nitrous oxide emissions from unmanaged livestock manure and urine on pastures, ranges, and paddocks, as well as from manure and urine that is spread onto fields is accounted for and discussed under Agricultural Soil Management.

Table 5-6, Table 5-7, and Table 5-8 (note, Table 5-8 is in units of gigagrams) provide estimates of methane and nitrous oxide emissions from manure management. Emission quantities are broken down by animal categories representing the major methane producing groups. Estimates for methane emissions in 1997 were 17.0 MMTCE (3.0 Tg). Emissions have increased each year from 1990 through 1995; however, emissions decreased slightly in 1996 with a decline in animal populations, including swine. In 1997, emissions from this source increased above even 1995 levels, mostly due to revived swine production and higher poultry production. Under the AgSTAR Program of the U.S. Climate Change Action Plan, methane emissions from manure have been reduced through methane recovery efforts. The AgSTAR Program reported a reduction of 0.1 MMTCE of methane in both 1996 and 1997.

Total N_2O emissions from managed manure systems in 1997 were estimated to be 3.0 MMTCE (35 Gg). The 15 percent increase in emissions from 1990 to 1997 can be attributed to an increase in the population of poultry and swine over the eight year period. The proportion of beef cattle in feedlots, which were assumed to use managed manure systems, also increased. Again, unmanaged livestock manure is accounted for under Agricultural Soil Management. Methane emissions were mostly unaffected by this shift in the beef cattle population because feedlot cattle use solid storage systems, which produce little methane.

In general, changes in the emission estimates over time reflect variations in animal populations. The estimates also reflect a regional redistribution of dairies to the southwestern states, which have larger average farm sizes, and an increase in feed consumption by dairy cows to accommodate increased milk production per cow. Regional shifts in the hog population were also assessed.

Methodology

The methods presented in EPA (1993) form the basis of the methane emissions estimates for each animal type. The calculation of emissions requires the following information:

- Amount of manure produced (amount per head times number of head)
- Portion of the manure that is volatile solids (by animal type)
- Methane producing potential of the volatile solids (by animal type)

Table 5-6: CH_4 and N_2O Emissions from Manure Management (MMTCE)

Gas/Animal Type	1990	1991	1992	1993	1994	1995	1996	1997
CH_4	**14.9**	**15.4**	**16.0**	**16.1**	**16.7**	**16.9**	**16.6**	**17.0**
Dairy Cattle	4.3	4.3	4.4	4.4	4.5	4.5	4.5	4.6
Beef Cattle	1.1	1.2	1.2	1.2	1.2	1.3	1.3	1.3
Swine	7.8	8.2	8.6	8.6	9.1	9.2	8.8	9.3
Sheep	+	+	+	+	+	+	+	+
Goats	+	+	+	+	+	+	+	+
Poultry	1.5	1.5	1.6	1.6	1.7	1.7	1.7	1.8
Horses	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2
N_2O	**2.6**	**2.8**	**2.8**	**2.9**	**2.9**	**2.9**	**3.0**	**3.0**
Dairy Cattle	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1
Beef Cattle	1.1	1.2	1.2	1.2	1.2	1.2	1.2	1.2
Swine	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1
Sheep	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Goats	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Poultry	1.3	1.3	1.4	1.4	1.5	1.5	1.5	1.6
Horses	0.0	0.0	0.0	0.0	0.1	0.1	0.1	0.1
Total	**17.6**	**18.2**	**18.7**	**19.0**	**19.7**	**19.8**	**19.5**	**20.0**

+ Does not exceed 0.05 MMTCE
Note: Totals may not sum due to independent rounding.

Table 5-7: CH_4 Emissions from Manure Management (Tg)

Animal Type	1990	1991	1992	1993	1994	1995	1996	1997
Dairy Cattle	0.7	0.8	0.8	0.8	0.8	0.8	0.8	0.8
Beef Cattle	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2
Swine	1.4	1.4	1.5	1.5	1.6	1.6	1.5	1.6
Sheep	+	+	+	+	+	+	+	+
Goats	+	+	+	+	+	+	+	+
Poultry	0.3	0.3	0.3	0.3	0.3	0.3	0.3	0.3
Horses	+	+	+	+	+	+	+	+
Total	**2.6**	**2.7**	**2.8**	**2.8**	**2.9**	**2.9**	**2.9**	**3.0**

+ Does not exceed 0.05 Tg
Note: Totals may not sum due to independent rounding.

Table 5-8: N_2O Emissions from Manure Management (Gg)

Animal Type	1990	1991	1992	1993	1994	1995	1996	1997
Dairy Cattle	1	1	1	1	1	1	1	1
Beef Cattle	13	15	14	15	15	14	14	15
Swine	1	1	1	1	1	1	1	1
Sheep	+	+	+	+	+	+	+	+
Goats	+	+	+	+	+	+	+	+
Poultry	15	16	16	17	17	18	18	19
Horses	1	1	1	1	1	1	1	1
Total	**31**	**33**	**33**	**34**	**35**	**34**	**35**	**36**

+ Does not exceed 0.5 Gg
Note: Totals may not sum due to independent rounding.

- Extent to which the methane producing potential is realized for each type of manure management system (by state and manure management system)
- Portion of manure managed in each manure management system (by state and animal type)

For dairy cattle and swine—the two largest emitters of methane—estimates were developed using state-level animal population data. For other animal types, 1990 emission estimates from the detailed analysis presented in EPA (1993) were scaled at the national level using the population of each livestock type. Nitrous oxide emissions were estimated by first determining manure management system usage. Manure system usage for dairy cows and swine were based on the farm size distribution. Total Kjeldahl nitrogen[4] production was calculated for all livestock using livestock population data and nitrogen excretion rates. The total amount of nitrogen from manure was reduced by 20 percent to account for the portion that volatilizes to NH_3 and NO_x (IPCC/UNEP/OECD/IEA 1997). Nitrous oxide emission factors were then applied to total nitrogen production to estimate N_2O emissions. Throughout the time series the estimates of the portion of manure and urine which is managed in each of the manure management systems in each state remained fixed.

See Annex H for more detailed information on the methodology and data used to calculate methane emissions from manure management. The same activity data was also used to calculate N_2O emissions.

Data Sources

Annual livestock population data for all livestock types except horses were obtained from the U.S. Department of Agriculture's National Agricultural Statistics Service (USDA 1994a, b; 1995a-j; 1996a-f; 1997a-f, 1998a-h). Horse population data were obtained from the FAOSTAT database (FAO 1998). Data on farm size distribution for dairy cows and swine were taken from the U.S. Department of Commerce (DOC 1995, 1987). Manure management system usage data for other livestock were taken from EPA (1992). Nitrogen excretion rate data were developed by the American Society of Agricultural Engineers (ASAE 1995). Nitrous oxide emission factors were taken from IPCC/UNEP/OECD/IEA (1997). Manure management systems characterized as "Other" generally refers to deep pit and litter systems. The IPCC N_2O emission factor for "other" systems (0.005 kg N_2O/kg N excreted), was determined to be inconsistent with the characteristics of these management systems. Therefore, in its place the solid storage/drylot emission factor was used.

Uncertainty

The primary factors contributing to the uncertainty in emission estimates are a lack of information on the usage of various manure management systems in each state and the exact methane generating characteristics of each type of manure management system. Because of significant shifts in the dairy and swine sectors toward larger farms, it is believed that increasing amounts of manure are being managed in liquid manure management systems. The existing estimates capture a portion of these shifts as the dairy and swine populations move regionally toward states with larger average farm sizes. However, changes in farm size distribution within states since 1992 are not captured by the method. The methane generating characteristics of each manure management system type are based on relatively few laboratory and field measurements, and may not match the diversity of conditions under which manure is managed nationally.

The N_2O emission factors published in IPCC/UNEP/OECD/IEA (1997) were also derived using limited information. The IPCC factors are global averages; U.S.-specific emission factors may be significantly different. Manure and urine in anaerobic lagoons and liquid/slurry management systems produce methane at different rates, and would in all likelihood produce N_2O at different rates, although a single emission factor was used.

Rice Cultivation

Most of the world's rice, and all rice in the United States, is grown on flooded fields. When fields are flooded, aerobic decomposition of organic material gradually depletes the oxygen present in the soil and

[4] Total Kjeldahl nitrogen is a measure of organically bound nitrogen and ammonia nitrogen.

floodwater causing anaerobic conditions in the soil to develop. Under such conditions, methane is produced through anaerobic decomposition of soil organic matter by methanogenic bacteria. However, not all of the methane that is produced is released into the atmosphere. As much as 60 to 90 percent of the methane produced is oxidized by aerobic methanotrophic bacteria in the soil (Holzapfel-Pschorn et al. 1985, Sass et al. 1990). Some of the methane is also leached away as dissolved methane in floodwater that percolates from the field. The remaining non-oxidized methane is transported from the submerged soil to the atmosphere primarily by diffusive transport through the rice plants. Some methane also escapes from the soil via diffusion and bubbling through floodwaters.

The water management system under which rice is grown is one of the most important factors affecting methane emissions. Upland rice fields are not flooded, and therefore are not believed to produce methane. In deepwater rice fields (i.e., fields with flooding depths greater than one meter), lower stems and roots of the rice plants are dead, and thus effectively block the primary methane transport pathway to the atmosphere. Therefore, while deepwater rice growing areas are believed to emit methane, the quantities released are likely to be significantly less than the quantities released from areas with more shallow flooding depths. Also, some flooded fields are drained periodically during the growing season, either intentionally or accidentally. If water is drained and soils are allowed to dry sufficiently, methane emissions decrease or stop entirely. This is due to soil aeration, which not only causes existing soil methane to oxidize but also inhibits further methane production in soils. All rice in the United States is grown under continuously flooded conditions; none is grown under deepwater conditions.

Other factors that influence methane emissions from flooded rice fields include soil temperature, soil type, fertilization practices, cultivar selection, and other cultivation practices (e.g., tillage, seeding and weeding practices). Many studies have found, for example, that methane emissions increase as soil temperature increases. Several studies have also indicated that some types of synthetic nitrogen fertilizer inhibit methane generation, while organic fertilizers enhance methane emissions. However, while it is generally acknowledged that these factors influence methane emissions, the extent of their influence, individually or in combination, has not been well quantified.

Rice cultivation is a small source of methane in the United States. Only seven states grow rice: Arkansas, California, Florida, Louisiana, Mississippi, Missouri, and Texas. Methane emissions from rice cultivation in 1997 were estimated to have been 2.7 MMTCE (475 Gg). Table 5-9 and Table 5-10 present annual emission estimates for each state. There was no apparent trend over the seven year period. Between 1994 and 1996, rice areas declined fairly steadily in almost all states, and the national total declined by about 8 percent each year; in 1997, however, rice areas increased by about 7 percent (see Table 5-11).

The factors that affect the rice area harvested vary from state to state. In Florida, the state having the smallest harvested rice area, rice acreage is driven by sugarcane acreage. Sugarcane fields are flooded each year to control pests, and on this flooded land a rice crop is grown along with a ratoon crop of sugarcane (Schudeman 1997a). In Missouri, rice acreage is affected by weather (rain during the planting season may prevent the planting of rice), prices of soybeans relative to rice (if soybean prices are higher, then soybeans may be planted on some of the land which would otherwise have been planted in rice), and government support programs (which, beginning in 1996, were being phased-out) (Stevens 1997). In Mississippi, rice acreage is driven by both the price of rice and the price of soybeans. Rice in Mississippi is usually rotated with soybeans, but if soybean prices increase relative to rice prices, then some of the acreage that would have been planted in rice, is instead planted in soybeans (Street 1997). In Texas, rice production, and thus, harvested area, are driven by both government programs and the cost of production (Klosterboer 1997). California rice area is influenced by water availability as well as government programs and commodity prices. In recent years, California was able to grow more rice due to recovery from a drought, as well as price increases associated with gaining access to the Japanese market (Scardaci 1997). In Louisiana, rice

Table 5-9: CH_4 Emissions from Rice Cultivation (MMTCE)

State	1990	1991	1992	1993	1994	1995	1996	1997
Arkansas	0.9	0.9	1.0	0.9	1.1	1.0	0.9	1.0
California	0.5	0.4	0.5	0.5	0.6	0.5	0.6	0.6
Florida	+	+	+	+	+	+	+	+
Louisiana	0.6	0.6	0.7	0.6	0.7	0.7	0.6	0.6
Mississippi	0.2	0.1	0.2	0.2	0.2	0.2	0.1	0.1
Missouri	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1
Texas	0.3	0.3	0.3	0.2	0.3	0.3	0.2	0.2
Total	**2.5**	**2.5**	**2.8**	**2.5**	**3.0**	**2.8**	**2.5**	**2.7**

+ Does not exceed 0.05 MMTCE
Note: Totals may not sum due to independent rounding.

Table 5-10: CH_4 Emissions from Rice Cultivation (Gg)

State	1990	1991	1992	1993	1994	1995	1996	1997
Arkansas	156	164	180	160	185	175	152	178
California	79	70	79	88	98	94	101	103
Florida	3	5	5	5	5	5	4	5
Louisiana	111	104	126	108	126	116	99	111
Mississippi	27	24	30	27	34	32	23	26
Missouri	11	12	15	12	16	15	12	14
Texas	52	50	51	43	52	46	40	38
Total	**439**	**429**	**486**	**443**	**516**	**482**	**431**	**475**

Note: Totals may not sum due to independent rounding.

area is influenced by government programs, weather conditions (such as rainfall during the planting season), as well as the price of rice relative to that of corn and other crops (Saichuk 1997). Arkansas rice area has been influenced in the past by government programs. The phase-out of these programs began in 1996, and commodity prices in the spring had a greater effect on the amount of land planted in rice (Mayhew 1997).

Methodology

The *Revised 1996 IPCC Guidelines* (IPCC/UNEP/OECD/IEA 1997) recommend applying a seasonal emission factor to the annual harvested rice area to estimate annual CH_4 emissions. This methodology assumes that a seasonal emission factor is available for all growing conditions, including season lengths. Because season lengths are variable both within and among states in the United States, and because flux measurements have not been taken under all growing conditions in the United States, the previous IPCC methodology (IPCC/UNEP/OECD/IEA 1995) has been applied here, using season lengths that vary slightly from the recommended approach. The 1995 *IPCC Guidelines* recommend multiplying a daily average emission factor by growing season length and annual harvested area. The *IPCC Guidelines* suggest that the “growing” season be used to calculate emissions based on the assumption that emission factors are derived from measurements over the whole growing season rather than just the flooding season. Applying this assumption to the United States, however, would result in an overestimate of emissions because the emission factors developed for the United States are based on measurements over the flooding, rather than the growing, season. Therefore, the method used here is based on the number of days of flooding during the growing season and a daily average emission factor, which is multiplied by the harvested area. Agricultural statisticians in each of the seven states in the United States that produce rice were contacted to determine water management practices and flooding season lengths in each state. Although all contacts reported that rice growing areas were continually flooded, flooding season lengths varied considerably among states; therefore, emissions were calculated separately for each state.

The climatic conditions of southwest Louisiana, Texas, and Florida also allow for a second, or ratoon, rice crop. This second rice crop is produced from regrowth on the stubble after the first crop has been harvested. The emission estimates presented here account for this additional harvested area.

Because the number of days that the rice fields remain permanently flooded varies considerably with planting system and cultivar type, a range for the flooding season length was adopted for each state. The harvested areas and flooding season lengths for each state are presented in Table 5-11 and Table 5-12, respectively.

Data Sources

Data on harvested rice area for all states except Florida were taken from U.S. Department of Agriculture's *Crop Production 1997 Summary* (USDA 1998). Harvested rice areas in Florida from 1990 to 1996 were obtained from Tom Schudeman (1997a), a Florida Agricultural Extension Agent. Harvested rice areas in Florida in 1997 were obtained from Terrie Smith of Sem-Chi Rice (1998). Acreages for the ratoon crops were estimated by assuming that the ratooned areas were equal to about 30 percent of the primary crop in Louisiana, 40 percent in Texas (Lindau and Bollich 1993); 50 percent of the primary crop in Florida in 1990 through 1996 (Schudeman 1995), and 67 percent of the primary crop in Florida in 1997 (Smith, 1998). Information about flooding season lengths was obtained from agricultural extension agents in every rice-producing state. Daily methane emission factors were taken from results of field studies performed in California (Cicerone et al. 1983), Texas (Sass et al. 1990, 1991a, 1991b, 1992) and Louisiana (Lindau et al. 1991, Lindau and Bollich 1993). Based on the maximal and minimal estimates of the emission rates measured in these studies, a range of 0.1065 to 0.5639 g/m^2/day was applied to the harvested areas and flooding season lengths in each state.[5] Since these measurements were taken in rice growing areas, they are representative of soil temperatures, and water and fertilizer management practices typical of the United States.

Uncertainty

There are three sources of uncertainty in the calculation of CH_4 emissions from rice cultivation. The largest uncertainty is associated with the emission factor. Daily average emissions, derived from field measurements in the United States, vary from state to state by as

Table 5-11: Area Harvested for Rice-Producing States (hectares)

State/Crop	1990	1991	1992	1993	1994	1995	1996	1997
Arkansas	485,633	509,915	558,478	497,774	574,666	542,291	473,493	554,431
California	159,854	141,071	159,450	176,851	196,277	188,183	202,347	206,394
Florida								
Primary	4,978	8,580	8,944	8,449	8,902	8,903	8,903	7,406
Ratoon	2,489	4,290	4,472	4,225	4,451	4,452	4,452	4,943
Louisiana								
Primary	220,558	206,394	250,911	214,488	250,911	230,676	215,702	221,773
Ratoon	66,168	61,918	75,273	64,346	75,273	69,203	64,711	66,532
Mississippi	101,174	89,033	111,291	99,150	126,669	116,552	84,176	96,317
Missouri	32,376	37,232	45,326	37,637	50,182	45,326	36,423	44,112
Texas								
Primary	142,857	138,810	142,048	120,599	143,262	128,693	120,599	104,816
Ratoon	57,143	55,524	56,819	48,240	57,305	51,477	48,240	41,926
Total	**1,273,229**	**1,255,767**	**1,413,011**	**1,271,759**	**1,487,897**	**1,385,755**	**1,259,045**	**1,348,650**

Note: Totals may not sum due to independent rounding.

[5] Two measurements from these studies were excluded when determining the emission coefficient range. A low seasonal average flux of 0.0595 g/m^2/day in Sass et al. (1990) was excluded because this site experienced a mid-season accidental drainage of floodwater, after which methane emissions declined substantially and did not recover for about two weeks. Also, the high seasonal average flux of 2.041 g/m^2/day in Lindau and Bollich (1993) was excluded since this emission rate is unusually high, compared to other flux measurements in the United States, as well as in Europe and Asia (see IPCC/UNEP/OECD/IEA 1997).

Table 5-12: Primary Cropping Flooding Season Length (days)

State	Low	High
Arkansas	75	100
California	123	153
Florida*	90	120
Louisiana*	90	120
Mississippi	75	82
Missouri	80	100
Texas*	60	80

* These states have a second, or "ratoon", cropping cycle which may have a shorter flooding season than the one listed in the table.

much as two orders of magnitude (IPCC/UNEP/OECD/IEA 1997). This variability is due to differences in cultivation practices, such as ratooning and fertilizer use, as well as differences in soil and climatic conditions. A range (0.3352 g/m^2/day ±68 percent) has been used in these calculations to reflect this variability. Based on this range, methane emissions from rice cultivation in 1997 were estimated to have been approximately 0.7 to 4.8 MMTCE (121 to 830 Gg).

Another source of uncertainty is in the flooding season lengths used for each state. Flooding seasons in each state may fluctuate from year to year and thus a range has been used to reflect this uncertainty.

The last source of uncertainty centers around the ratoon, or second crop. Rice fields for the ratoon crop typically remain flooded for a shorter period of time than for the first crop. Studies indicate, however, that the methane emission rate of the ratoon crop may be significantly higher than that of the first crop. The rice straw produced during the first harvest has been shown to dramatically increase methane emissions during the ratoon cropping season (Lindau and Bollich 1993). It is not clear to what extent the shorter season length and higher emission rates offset each other. As scientific understanding improves, these emission estimates can be adjusted to better reflect these variables.

Agricultural Soil Management

Nitrous oxide (N_2O) is produced naturally in soils through the microbial processes of nitrification and denitrification.[6] A number of agricultural activities add nitrogen to soils, thereby increasing the amount of nitrogen available for nitrification and denitrification, and ultimately the amount of N_2O emitted. These activities may add nitrogen to soils either directly or indirectly. Direct additions occur through various cropping practices (i.e., application of synthetic and organic fertilizers, application of animal wastes, production of nitrogen-fixing crops, incorporation of crop residues, and cultivation of high organic content soils, called histosols), and through animal grazing (i.e., direct deposition of animal wastes on pastures, range, and paddocks by grazing animals). Indirect additions occur through two mechanisms: 1) volatilization of applied nitrogen (i.e., fertilizer and animal waste) and subsequent atmospheric deposition of that nitrogen as ammonia (NH_3) and oxides of nitrogen (NO_x); and 2) surface runoff and leaching of applied nitrogen. Other agricultural soil management practices, such as irrigation, drainage, tillage practices, and fallowing of land, can affect fluxes of N_2O, as well as other greenhouse gases, to and from soils. However, because there are significant uncertainties as to the effects of these other practices, they have not been estimated.

Estimates of annual N_2O emissions from agricultural soil management were underestimated in the previous U.S. Inventory because the animal waste portion of direct N_2O emissions from agricultural cropping practices included only animal wastes managed as "daily spread." However, of the total animal waste nitrogen produced in the U.S., all of it (i.e., nitrogen from animal wastes managed as daily spread and managed in animal waste management systems) will eventually be applied to soils with the exception of that which volatilizes, runs off, is used for feed, and is directly deposited by grazing animals. The present inventory accounts for total animal waste nitrogen, and, as a consequence, the emission esti-

[6] Nitrification is the aerobic microbial oxidation of ammonium to nitrate, and denitrification is the anaerobic microbial reduction of nitrate to dinitrogen gas (IPCC/UNEP/OECD/IEA 1997). Nitrous oxide is a gaseous intermediate product in the reaction sequences of both processes, which leaks from microbial cells into the soil atmosphere.

mates provided below are higher (by about five percent) than previous estimates.

The revised estimates of annual N_2O emissions from agricultural soil management range from 65.3 to 74.1 MMTCE (773 to 876 Gg N_2O) for the years 1990 to 1997 (Table 5-13 and Table 5-14). Emission levels increased fairly steadily from 1990 to 1997 except for the year 1993, when emissions declined slightly, and the year 1994, when emissions increased sharply. These fluctuations are largely a reflection of annual variations in synthetic nitrogen fertilizer consumption and crop production. Synthetic nitrogen fertilizer consumption, and production of corn and most beans and pulses, peaked in 1994 due to the 1993 flooding of the North Central region and the intensive cultivation that followed. Over the eight-year period, total emissions of N_2O increased by 13 percent.

Methodology and Data Sources

This N_2O source category is divided into three components: (1) direct emissions from agricultural soils due to cropping practices; (2) direct emissions from agricultural soils due to grazing animals; and (3) emissions from soils indirectly induced by agricultural applications of nitrogen. The emission estimates for all three components follow the methodologies in the *Revised 1996 IPCC Guidelines* (IPCC/UNEP/OECD/IEA 1997).

Direct N_2O Emissions from Agricultural Soils

Estimates of N_2O emissions from this component are based on the total amount of nitrogen that is applied to soils through cropping practices. These practices are (1) the application of synthetic and organic fertilizers, (2) the application of animal waste through both daily spread and eventual application of wastes that had been managed in waste management systems (e.g., lagoons), (3) the production of nitrogen-fixing crops, (4) the incorporation of crop residues into the soil, and (5) the cultivation of histosols.

Annual synthetic and organic fertilizer consumption data for the U.S. were taken from annual publications on commercial fertilizer statistics (AAPFCO 1995, 1996, 1997; TVA 1990, 1992a,b, 1994). Organic fertilizers included in these publications are manure, compost, dried blood, sewage sludge, tankage[7], and other organic. The manure portion of the organic fertilizers was subtracted from the total organic fertilizer consumption data to avoid double counting[8]. Fertilizer consumption data are recorded in "fertilizer

Table 5-13: N_2O Emissions from Agricultural Soil Management (MMTCE)

Activity	1990	1991	1992	1993	1994	1995	1996	1997
Direct								
Agricultural Soils	36.5	37.0	38.4	36.7	42.1	39.0	40.8	43.0
Grazing Animals	10.1	10.1	10.4	10.5	10.8	11.0	10.8	10.7
Indirect	18.8	19.1	19.2	19.7	20.4	20.1	20.4	20.4
Total	**65.3**	**66.2**	**68.0**	**67.0**	**73.4**	**70.2**	**72.0**	**74.1**

Note: Totals may not sum due to independent rounding.

Table 5-14: N_2O Emissions from Agricultural Soil Management (Gg N_2O)

Activity	1990	1991	1992	1993	1994	1995	1996	1997
Direct								
Agricultural Soils	431	438	454	434	498	461	482	509
Grazing Animals	119	120	123	125	128	131	128	126
Indirect	222	225	227	233	241	238	241	241
Total	**773**	**783**	**804**	**792**	**868**	**830**	**851**	**876**

Note: Totals may not sum due to independent rounding.

[7] Tankage is dried animal residue, usually freed from fat and gelatin.

[8] The manure is accounted for when estimating the total amount of nitrogen from manure applied to soils.

year" totals (July to June) which were converted to calendar year totals by assuming that approximately 35 percent of fertilizer usage occurred from July to December (TVA 1992b). July to December values were not available for calendar year 1997, so a "least squares line" statistical test using the past seven data points was used to arrive at an approximate total. Data on the nitrogen content of synthetic fertilizers were available in published consumption reports; however, data on non-manure organic fertilizer consumption did not include nitrogen content information. To convert to units of nitrogen, it was assumed that 4.1 percent of non-manure organic fertilizers (on a mass basis) was nitrogen (Terry 1997). Annual consumption of commercial fertilizers (synthetic and non-manure organic) in units of nitrogen are presented in Table 5-15. The total amount of nitrogen consumed from synthetic and non-manure organic fertilizers was reduced by 10 percent and 20 percent, respectively, to account for the portion that volatilizes to NH_3 and NO_x (IPCC/UNEP/OECD/IEA 1997).

To estimate the amount of animal waste nitrogen applied to soils, it was assumed that of the total animal waste nitrogen produced in the U.S., all of it will eventually be applied to soils with three exceptions. These exceptions are (1) the portion of nitrogen that will volatilize, (2) the nitrogen in the poultry waste that is used as feed for ruminants (i.e., approximately 10% of the poultry waste produced in the U.S.), and (3) the nitrogen in the waste that is directly deposited onto fields by grazing animals[9]. Annual animal population data for all livestock types, except horses, were obtained from the USDA National Agricultural Statistics Service (USDA 1994b,c, 1995a-j, 1996a-g, 1997a-g, 1998a-g). Horse population data were taken from U.S. Department of Commerce's Bureau of Census (DOC 1987) and FAO (1996). Population data (by animal type) were multiplied by an average animal mass constant (ASAE 1995) to derive total animal mass for each animal type. Total Kjeldahl nitrogen[10] excreted per year (manure and urine) was then calculated using daily rates of N excretion per unit of animal mass (ASAE 1995) (see Table 5-16). The amount of animal waste nitrogen directly deposited by grazing animals, derived using manure management system usage data and farm size (Safely et al. 1992, DOC 1995) as described in the "Direct N_2O Emissions from Grazing Animals" section, was then subtracted from the total nitrogen. Ten percent of the poultry waste nitrogen produced in managed systems and used as feed for ruminants was then subtracted. Finally, the total amount of nitrogen from manure applied to soils was then reduced by 20 percent to account for the portion that volatilizes to NH_3 and NO_x (IPCC/UNEP/OECD/IEA 1997).

Annual production statistics for nitrogen-fixing crops (beans, pulses, and alfalfa) were taken from U.S. Department of Agriculture reports (USDA 1994a, 1997h, 1998h). These statistics are presented in Table 5-17. Crop product values for beans and pulses were expanded to total crop dry biomass, in mass units of dry matter, by applying residue to crop ratios and dry matter fractions for residue from Strehler and Stützle (1987). Crop product values for the alfalfa were converted to dry matter mass units by applying a dry matter fraction value estimated at 80 percent (Mosier 1998). To convert to units of nitrogen, it was assumed that 3 percent of the total crop dry mass for all crops was nitrogen (IPCC/UNEP/OECD/IEA 1997).

To estimate the amount of nitrogen applied to soils through crop residue incorporation, it was assumed that all residues from corn, wheat, bean, and pulse production, except the fractions that are burned in the field after harvest, are plowed under. Annual production statistics were taken from U.S. Department of Agriculture (USDA 1994a, 1997h, 1998h). These statistics are presented in Table 5-17 and Table 5-18. Crop residue biomass, in dry matter mass units, was calculated from the production statistics by applying residue to crop mass ratios and dry matter fractions for residue from Strehler and Stützle (1987). For wheat and corn, nitrogen contents were taken from Barnard and Kristoferson (1985). For beans and pulses, it was assumed that 3 percent of the total crop residue was nitrogen (IPCC/UNEP/OECD/IEA 1997).

[9] An additional exception is the nitrogen in the waste that will runoff from waste management systems due to inadequate management. There is insufficient information with which to estimate this fraction of waste nitrogen.

[10] Total Kjeldahl nitrogen is a measure of organically bound nitrogen and ammonia nitrogen.

The crops whose residues were burned in the field are corn, wheat, soybeans, and peanuts. For these crop types, the total residue nitrogen was reduced by 3 percent to subtract the fractions burned in the field (see the Agricultural Residue Burning section of this chapter).

Total crop nitrogen in the residues returned to soils was then added to the unvolatilized applied nitrogen from commercial fertilizers and animal wastes, and the nitrogen fixation from bean, pulse, and alfalfa cultivation. The sum was multiplied by the IPCC default emission factor (0.0125 kg N_2O-N/kg N applied) to estimate annual N_2O emissions from nitrogen applied to soils.

Statistics on the area of histosols cultivated annually were not available, so an estimate for the year 1982 (Mausbach and Spivey 1993) was used for all years in the 1990 to 1997 series (see Table 5-19). The area estimate was derived from USDA land-use statistics. The histosol area cultivated was multiplied by the IPCC default emission factor (5 kg N_2O-N/ha cultivated) to estimate annual N_2O emissions from histosol cultivation.

Annual N_2O emissions from nitrogen applied to soils were then added to annual N_2O emissions from histosol cultivation to estimate total direct annual N_2O emissions from agricultural cropping practices (see Table 5-20).

Direct N_2O Emissions from Grazing Animals

Estimates of N_2O emissions from this component were based on animal wastes that are not used as animal feed, or applied to soils, or managed in manure management systems, but instead are deposited directly on soils by animals in pastures, range, and paddocks.[11] It was assumed that all unmanaged wastes, except for dairy cow wastes, fall into this category (Safely et al. 1992). Estimates of nitrogen excretion by these animals were derived from animal population and weight statistics, information on manure management system usage in the United States, and nitrogen excretion values for each animal type.

Annual animal population data for all livestock types, except horses, were obtained from the USDA National Agricultural Statistics Service (USDA 1994b,c, 1995a-j, 1996a-g, 1997a-g, 1998a-g). Horse population data were taken from U.S. Department of Commerce's Bureau of Census (DOC 1987) and FAO (1996). Manure management system usage for all livestock types, except swine, was taken from Safely et al. (1992). Because these data were not available for swine, the swine population values were allocated to manure management system types using information on farm size distribution reported by the U.S. Department of Commerce (DOC 1995). Swine populations in the larger farm categories were assumed to utilize manure collection and storage management systems; all the wastes from smaller farms were assumed to be managed as pasture, range, and paddock. Population data for animals whose wastes were managed in pasture, range, and paddock were multiplied by an average animal mass constant (ASAE 1995) to derive total animal mass for each animal type. Total Kjeldahl nitrogen excreted per year was then calculated for each animal type using daily rates of N excretion per unit of animal mass (ASAE 1995). Annual nitrogen excretion was then summed over all animal types (see Table 5-16), and reduced by 20 percent to account for the portion that volatilizes to NH_3 and NO_x. The remainder was multiplied by the IPCC default emission factor (0.02 kg N_2O-N/kg N excreted) to estimate N_2O emissions (see Table 5-21).

Indirect N_2O Emissions from Nitrogen Applied to Agricultural Soils

This component accounts for N_2O that is emitted indirectly from nitrogen applied as fertilizer and excreted by livestock. Through volatilization, some of this nitrogen enters the atmosphere as NH_3 and NO_x, and subsequently returns to soils through atmospheric deposition, thereby enhancing N_2O production. Additional nitrogen is lost from soils through leaching and runoff, and enters groundwater and surface water systems, from which a portion is emitted as N_2O. These two indirect emission pathways are treated separately, although the activity data used are identical.

[11] The *Revised 1996 IPCC Guidelines* (IPCC/UNEP/OECD/IEA 1997) indicate that emissions from animal wastes managed in solid storage and drylot should also be included in the emissions from soils (see footnote "c" in Table 4-22 in the Reference Manual); however, this instruction appeared to be an error (and footnote "b" should have been listed next to "Solid storage and drylot" in Table 4-22). Therefore, N_2O emissions from livestock wastes managed in solid storage and drylot are reported under Manure Management, rather than here. (See Annex H for a discussion of the activity data used to calculate emissions from the manure management source category.)

Estimates of total nitrogen applied as fertilizer and excreted by all livestock (i.e., wastes from all unmanaged and managed systems) were derived using the same approach as was employed to estimate the direct soil emissions. Annual application rates for synthetic and non-manure organic fertilizer nitrogen were derived as described above from commercial fertilizer statistics for the United States (AAPFCO 1995, 1996, 1997; TVA 1990, 1992a and b, 1994). Annual total nitrogen excretion data (by animal type) were derived, also as described above, using animal population statistics (USDA 1994b,c, 1995a-j, 1996a-g, 1997a-g, 1998a-g; DOC 1987, 1998a,b, d-h; and FAO 1996), average animal mass constants (ASAE 1995), and daily rates of N excretion per unit of animal mass (ASAE 1995). Annual nitrogen excretion was then summed over all animal types.

To estimate N_2O emissions from volatilization and subsequent atmospheric deposition, it was assumed that 10 percent of the synthetic fertilizer nitrogen applied, 20 percent of the non-manure organic fertilizer nitrogen applied, and 20 percent of the total livestock nitrogen excretion were volatilized to NH_3 and NO_x, and 1 percent of the total volatilized nitrogen returned to the soils and was emitted as N_2O (IPCC/UNEP/OECD/IEA 1997). These emission levels are presented in Table 5-22.

To estimate N_2O emissions from leaching and runoff, it was assumed that 30 percent of the non-volatilized nitrogen applied or excreted (i.e., 30 percent of the sum of 90 percent of synthetic fertilizer nitrogen plus 80 percent of non-manure organic fertilizer nitrogen plus 80 percent of total livestock nitrogen) was lost to leaching and surface runoff, and 2.5 percent of the lost nitrogen was emitted as N_2O (IPCC/UNEP/OECD/IEA 1997). These emission levels are also presented in Table 5-22.

Uncertainty

A number of conditions can affect nitrification and denitrification rates in soils, including: water content, which regulates oxygen supply; temperature, which controls rates of microbial activity; nitrate or ammonium concentration, which regulate reaction rates; available organic carbon, which is required for microbial activity; and soil pH, which is a controller of both nitrification and denitrification rates and the ratio of N_2O/N_2 from denitrification. These conditions vary greatly by soil type, climate, cropping system, and soil management regime. Although numerous emissions measurement data have been collected under a wide variety of controlled conditions, the interaction of these conditions and their combined effect on the processes leading to N_2O emissions are not fully understood. Moreover, the amount of added nitrogen from each source (fertilizers, animal wastes, nitrogen fixation, crop residues, cultivation of histosols, atmospheric deposition, or leaching and runoff) that is not absorbed by crops or wild vegetation, but remains in the soil and is available for production of N_2O, is uncertain. Therefore, it is not yet possible to develop statistically valid estimates of emission factors for all possible combinations of soil, climate, and management conditions. The emission factors used were midpoint estimates based on measurements described in the scientific literature, and as such, are representative of current scientific understanding. Nevertheless, estimated ranges around each midpoint estimate are wide; most are an order of magnitude or larger (IPCC/UNEP/OECD/IEA 1997).

Uncertainties also exist in the activity data used to derive emission estimates. In particular, the fertilizer statistics include only those organic fertilizers that enter the commercial market, so any non-commercial fertilizer use (other than livestock waste and incorporation of crop residues) has not been captured. For example, sewage sludge applied to soils (other than the portion in commercial organic fertilizers) has not been accounted for. Also, the nitrogen content of organic fertilizers varies by type, as well as within individual types; however, average values were used to estimate total organic fertilizer nitrogen consumed. Conversion factors for the bean, pulse, and alfalfa production statistics were based on a limited number of studies, and may not be representative of all conditions in the United States. It was assumed that the entire crop residue for corn, wheat, beans, and pulses was returned to the soils, with the exception of the fraction burned. A portion of this residue may be disposed of through other practices, such as composting or landfilling; however, data on these practices are not available. Statistics on the histosol area cultivated annually were not available either; the point estimate reported should be considered highly uncertain. Lastly, the live-

Table 5-15: Commercial Fertilizer Consumption (Metric Tons of Nitrogen)

Fertilizer Type	1990	1991	1992	1993	1994	1995	1996	1997
Synthetic	10,110,726	10,271,698	10,335,778	10,727,695	11,171,243	10,811,665	11,164,582	11,214,037
Non-Manure Organics	763	1,210	1,256	1,121	1,101	1,368	1,533	1,534

Note: These figures do not include manure used as commercial fertilizer.

Table 5-16: Animal Excretion (Metric Tons of Nitrogen)

Activity	1990	1991	1992	1993	1994	1995	1996	1997
Applied to Soils	3,062,628	3,150,736	3,135,107	3,158,899	3,215,171	3,185,729	3,167,180	3,194,314
Pasture, Range, & Paddock	4,742,247	4,761,332	4,881,526	4,952,799	5,095,799	5,192,152	5,099,376	5,022,867
All Management Systems	7,865,794	7,975,050	8,081,690	8,178,644	8,379,974	8,448,804	8,339,367	8,291,710

Table 5-17: Bean, Pulse, and Alfalfa Production (Metric Tons of Product)

Product Type	1990	1991	1992	1993	1994	1995	1996	1997
Soybeans	52,415,690	54,064,730	59,611,670	50,919,130	69,625,980	59,243,170	64,837,320	74,223,690
Peanuts	1,634,590	2,234,650	1,943,380	1,538,770	1,934,370	1,570,100	1,660,690	1,608,600
Dry Edible Beans	1,468,690	1,531,550	1,025,800	993,960	1,323,900	1,397,610	1,268,240	1,332,490
Dry Edible Peas	107,590	168,510	114,990	149,320	102,290	209,060	121,150	263,810
Austrian Winter Peas	5,760	6,300	4,490	7,030	2,310	5,400	4,670	5,220
Lentils	66,459	104,090	71,030	90,990	84,190	97,300	60,460	108,450
Wrinkled Seed Peas	41,820	41,960	24,360	38,510	34,200	47,540	24,860	30,940
Alfalfa	75,671,002	75,585,727	71,794,602	72,851,472	73,786,780	76,670,720	72,136,611	71,887,135

Table 5-18: Corn and Wheat Production (Metric Tons of Product)

Product Type	1990	1991	1992	1993	1994	1995	1996	1997
Corn for Grain	201,533,597	189,867,775	240,719,220	160,953,750	256,621,290	187,305,080	236,064,120	237,896,540
Wheat	74,292,383	53,890,553	67,135,240	65,220,410	63,166,750	59,400,390	62,191,130	68,761,480

Table 5-19: Histosol Area Cultivated (Hectares)

Year	Hectares
1990	843,386
1991	843,386
1992	843,386
1993	843,386
1994	843,386
1995	843,386
1996	843,386
1997	843,386

Table 5-20: Direct N_2O Emissions from Agricultural Cropping Practices (MMTCE)

Activity	1990	1991	1992	1993	1994	1995	1996	1997
Commercial Fertilizers (excluding manure)	15.1	15.4	15.5	16.0	16.7	16.2	16.7	16.8
Animal Waste Applied to Soils	4.1	4.2	4.2	4.2	4.3	4.2	4.2	4.2
N Fixation	10.3	10.6	11.1	9.9	12.5	11.3	11.8	13.1
Crop Residue	6.4	6.3	7.1	6.0	8.0	6.8	7.5	8.4
Histosol Cultivation	0.6	0.6	0.6	0.6	0.6	0.6	0.6	0.6
Total	**36.5**	**37.0**	**38.4**	**36.7**	**42.1**	**39.0**	**40.8**	**43.0**

Note: Totals may not sum due to independent rounding.

Table 5-21: Direct N_2O Emissions from Pasture, Range, and Paddock Animals (MMTCE)

Animal Type	1990	1991	1992	1993	1994	1995	1996	1997
Beef Cattle	9.0	9.1	9.3	9.5	9.8	10.0	9.8	9.7
Horses	0.5	0.5	0.6	0.6	0.6	0.6	0.6	0.6
Swine	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2
Sheep	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2
Goats	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.0
Poultry	+	+	+	+	+	+	+	+
Total	**10.1**	**10.1**	**10.4**	**10.5**	**10.8**	**11.0**	**10.8**	**10.7**

+ Does not exceed 0.05 MMTCE
Note: Totals may not sum due to independent rounding.

Table 5-22: Indirect N_2O Emissions (MMTCE)

Activity	1990	1991	1992	1993	1994	1995	1996	1997
Volatilization & Atmospheric Deposition	**3.4**	**3.5**	**3.5**	**3.6**	**3.7**	**3.7**	**3.7**	**3.7**
Commercial Fertilizer (excluding manure)	1.3	1.4	1.4	1.4	1.5	1.4	1.5	1.5
Animal Waste	2.1	2.1	2.1	2.2	2.2	2.2	2.2	2.2
Surface Run-off & Leaching	**15.3**	**15.6**	**15.7**	**16.1**	**16.7**	**16.4**	**16.7**	**16.7**
Commercial Fertilizer (excluding manure)	9.1	9.2	9.3	9.6	10.0	9.7	10.0	10.1
Animal Waste	6.3	6.4	6.4	6.5	6.7	6.7	6.6	6.6
Total	**18.8**	**19.1**	**19.2**	**19.7**	**20.4**	**20.1**	**20.4**	**20.4**

Note: Totals may not sum due to independent rounding.

stock excretion values, while based on detailed population and weight statistics, were derived using simplifying assumptions concerning the types of management systems employed.

Agricultural Residue Burning

Large quantities of agricultural crop residues are produced by farming activities. There are a variety of ways to dispose of these residues. For example, agricultural residues can be plowed back into the field, composted, landfilled, or burned in the field. Alternatively, they can be collected and used as a fuel or sold in supplemental feed markets. Field burning of crop residues is not considered a net source of carbon dioxide (CO_2) because the carbon released to the atmosphere as CO_2 during burning is assumed to be reabsorbed during the next growing season. Crop residue burning is, however, a net source of methane (CH_4), nitrous oxide (N_2O), carbon monoxide (CO), and nitrogen oxides (NO_x), which are released during combustion. In addition, field burning may result in enhanced emissions of N_2O and NO_x many days after burning (Anderson *et al*. 1988, Levine *et al*. 1988), although this process is highly uncertain and was not quantified.

Field burning is not a common method of agricultural residue disposal in the United States; therefore, emissions from this source are minor. The primary crop types whose residues are typically burned in the United States are wheat, rice, sugarcane, peanut, soybeans, barley, and corn, and of these residues, generally less than 5 percent is burned each year.[12] Annual emissions from this source over the period 1990 through 1997 averaged approximately 0.2 MMTCE (37 Gg) of CH_4, 0.1 MMTCE (1 Gg) of N_2O, 771 Gg of CO, and 32 Gg of NO_x (see Table 5-23 and Table 5-24). The average annual emission estimates for field burning of crop residues from 1990 through 1997 represent approximately 1 percent of total U.S. CO emissions.

Methodology

The methodology for estimating greenhouse gas emissions from field burning of agricultural residues is consistent with the *Revised 1996 IPCC Guidelines* (IPCC/UNEP/OECD/IEA 1997). In order to estimate the amounts of carbon and nitrogen released during burning, the following equations were used:

Carbon Released = (Annual Crop Production) × (Residue/Crop Product Ratio) × (Fraction of Residues Burned *in situ*) × (Dry Matter content of the Residue) × (Burning Efficiency) × (Carbon Content of the Residue) × (Combustion Efficiency)[13]

Nitrogen Released = (Annual Crop Production) × (Residue/Crop Product Ratio) × (Fraction of Residues Burned *in situ*) × (Dry Matter Content of the Residue) × (Burning Efficiency) × (Nitrogen Content of the Residue) × (Combustion Efficiency)

Emissions of CH_4 and CO were calculated by multiplying the amount of carbon released by the appropriate emission ratio (i.e., CH_4/C or CO/C). Similarly, N_2O and NO_x emissions were calculated by multiplying the amount of nitrogen released by the appropriate emission ratio (i.e., N_2O/N or NO_x/N).

Table 5-23: Emissions from Agricultural Residue Burning (MMTCE)

Gas/Crop Type	1990	1991	1992	1993	1994	1995	1996	1997
CH_4	**0.2**	**0.2**	**0.2**	**0.2**	**0.2**	**0.2**	**0.2**	**0.2**
Wheat	+	+	+	+	+	+	+	+
Rice	+	+	+	+	+	+	+	+
Sugarcane	+	+	+	+	+	+	+	+
Corn	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1
Barley	+	+	+	+	+	+	+	+
Soybeans	+	+	+	+	+	+	+	0.1
Peanuts	+	+	+	+	+	+	+	+
N_2O	**0.1**	**0.1**	**0.1**	**0.1**	**0.1**	**0.1**	**0.1**	**0.1**
Wheat	+	+	+	+	+	+	+	+
Rice	+	+	+	+	+	+	+	+
Sugarcane	+	+	+	+	+	+	+	+
Corn	+	+	+	+	+	+	+	+
Barley	+	+	+	+	+	+	+	+
Soybeans	0.1	0.1	0.1	0.0	0.1	0.1	0.1	0.1
Peanuts	+	+	+	+	+	+	+	+
Total	**0.3**	**0.3**	**0.3**	**0.3**	**0.4**	**0.3**	**0.3**	**0.4**

+ Does not exceed 0.05 MMTCE
Note: Totals may not sum due to independent rounding.

[12] The fraction of rice straw burned each year is thought to be significantly higher (see "Data Sources" discussion below).

[13] Burning Efficiency is defined as the fraction of dry biomass exposed to burning that actually burns. Combustion Efficiency is defined as the fraction of carbon in the fire that is oxidized completely to CO_2. In the methodology recommended by the IPCC, the "burning efficiency" is assumed to be contained in the "fraction of residues burned" factor. However, the number used here to estimate the "fraction of residues burned" does not account for the fraction of exposed residue that does not burn. Therefore, a "burning efficiency factor" was added to the calculations.

Table 5-24: Emissions from Agricultural Residue Burning (Gg)

Gas/Crop Type	1990	1991	1992	1993	1994	1995	1996	1997
CH_4	**36**	**34**	**39**	**32**	**41**	**33**	**37**	**40**
Wheat	7	5	6	6	6	5	5	6
Rice	4	4	5	4	4	3	3	3
Sugarcane	1	1	1	1	1	1	1	1
Corn	17	16	19	14	20	16	19	19
Barley	1	1	1	1	1	1	1	1
Soybeans	7	7	8	7	9	8	9	10
Peanuts	+	+	+	+	+	+	+	+
N_2O	**1**	**1**	**1**	**1**	**2**	**1**	**1**	**2**
Wheat	+	+	+	+	+	+	+	+
Rice	+	+	+	+	+	+	+	+
Sugarcane	+	+	+	+	+	+	+	+
Corn	+	+	1	+	1	+	1	1
Barley	+	+	+	+	+	+	+	+
Soybeans	1	1	1	1	1	1	1	1
Peanuts	+	+	+	+	+	+	+	+
NO_x	**30**	**30**	**34**	**28**	**37**	**30**	**34**	**37**
Wheat	1	1	1	1	1	1	1	1
Rice	3	3	3	3	3	2	2	2
Sugarcane	+	+	+	+	+	+	+	0
Corn	11	11	13	9	14	10	13	13
Barley	+	+	+	+	+	+	+	0
Soybeans	14	14	16	14	18	16	17	20
Peanuts	+	+	+	+	+	+	+	0
CO	**763**	**712**	**824**	**681**	**858**	**703**	**786**	**843**
Wheat	137	99	124	120	116	109	114	127
Rice	88	88	89	84	86	65	59	65
Sugarcane	18	20	20	20	20	20	19	20
Corn	354	333	404	296	425	326	393	406
Barley	15	16	16	14	13	13	14	13
Soybeans	148	153	168	144	194	167	183	210
Peanuts	2	3	3	2	3	2	2	2

+ Does not exceed 0.5 Gg
Note: Totals may not sum due to independent rounding.

Data Sources

The crop residues burned in the United States were determined from various state level greenhouse gas emission inventories (ILENR 1993, Oregon Department of Energy 1995, Wisconsin Department of Natural Resources 1993) and publications on agricultural burning in the United States (Jenkins et al. 1992, Turn et al. 1997, EPA 1992). Crop production data were taken from the USDA's *Crop Production Summaries* (USDA 1993, 1994, 1995, 1996, 1997, 1998), except data on the production of rice in Florida. Data for the years 1996 and 1997 were obtained from Ken Vaodivia (1997) and Terrie Smith (1998) respectively, of Sem-Chi Rice. Rice production data were not available for the years 1990 to 1995, so they were estimated by applying the 1997 ratio of Florida rice production to Florida rice area to the total Florida rice area (both primary and ratoon) for 1990 to 1995. The 1990 to 1995 Florida rice areas were obtained from Tom Schudeman (1997), a Florida Agricultural Extension Agent. The percentage of crop residue burned was assumed to be 3 percent for all crops, except rice, based on state inventory data (ILENR 1993, Oregon Department of Energy 1995, Noller 1996, Wisconsin Department of Natural Resources 1993, and Cibrowski 1996). For rice, the only data that were available on percentage of crop residue burned were for California (Jenkins 1997), which was responsible for about 21 percent of the annual U.S. rice production. Until 1991, 99 percent of California's rice area was burned each year after harvest. Since then, California has tightened restrictions on burning, such that today, only about half of its rice area is burned each year. Therefore, a weighted average fraction burned was calculated for rice for each year assuming that the fraction of rice residue burned in California declined linearly from 99 to 50 percent between 1991 and 1996, and remained constant at 50 percent in 1997, while the fraction burned in the rest of the country stayed constant at 3 percent.

Residue/crop product ratios, residue dry matter contents, residue carbon contents, and residue nitrogen contents for all crops except sugarcane, peanuts, and soybeans were taken from Strehler and Stützle (1987). These data for sugarcane were taken from University of California (1977) and Turn et al. (1997). Residue/crop product ratios and residue dry matter contents for peanuts and soybeans were taken from Strehler and Stützle (1987); residue carbon contents for these crops were set at 0.45 and residue nitrogen contents were taken from Barnard and Kristoferson (1985) (the value for peanuts was set equal to the soybean value). The burning efficiency was assumed to be 93 percent, and the combustion efficiency was assumed to be 88 percent for all crop types (EPA 1994). Emission ratios for all gases were taken from the *Revised 1996 IPCC Guidelines* (IPCC/UNEP/OECD/IEA 1997).

Uncertainty

The largest source of uncertainty in the calculation of non-CO_2 emissions from field burning of agricultural residues is in the estimates of the fraction of residue of each crop type burned each year. Data on the fraction burned, or even the gross amount of residue burned each year, are not collected at either the national or state level. In addition, burning practices are highly variable among crops, as well as among states. The fractions of residue burned used in these calculations were based upon information collected by state agencies and in published literature. It is likely that these emission estimates will continue to change as more information becomes available.

Other sources of uncertainty include the residue/crop product ratios, residue dry matter contents, burning and combustion efficiencies, and emission ratios. A residue/crop product ratio for a specific crop can vary among cultivars, and for all crops except sugarcane, generic residue/crop product ratios, rather than ratios specific to the United States, have been used. Residue dry matter contents, burning and combustion efficiencies, and emission ratios, all can vary due to weather and other combustion conditions, such as fuel geometry. Values for these variables were taken from literature on agricultural biomass burning.